Title: A spatial approach to identify priority areas for pesticide pollution mitigation

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A spatial approach to identify priority areas for pesticide pollution mitigation

1. INTRODUCTION

Pesticide residues frequently occur in surface waters in Europe (Reemtsma et al., 2013; VMM, 2017, 2015) potentially having an impact on aquatic organisms or communities (Lefrancq et al., 2017). Treatment or targeted mitigation can prevent pesticide pollution from dispersing in the environment (Gregoire et al., 2009). Therefore, there is a need to implement mitigation measures in agriculture to ensure food production while reducing the environmental impact of pesticides and simultaneously achieving good water quality (European Commission, 2000, 2009).

Agricultural non-point-source (NPS) pesticide pollution is defined as inputs along the entire watercourse from applications of agrochemicals onto farmland (Holvoet et al., 2007). Once pesticides are applied and released in the environment, their fate is affected by their physical and chemical properties and the interactions with soil (Borggaard and Gimsing, 2008; Chaplain et al., 2011; Maqueda et al., 2017), climatology (Doppler et al., 2014; Leu et al., 2004a, 2004b), and agricultural practices (Alletto et al., 2010; Potter et al., 2015). The most important routes of diffuse pesticide pollution in water bodies are surface runoff and soil erosion, drain flow, leaching and spray drift (Reichenberger et al., 2007; Tang et al., 2012). Knowledge of these pathways and their relative importance is a prerequisite for developing mitigation strategies for polluted surface water (Bereswill et al., 2014; Holvoet et al., 2007; Reichenberger et al., 2007; Tang et al., 2012).

Areas within a catchment pose varying risks of pollution. Critical source areas (CSAs) contribute a considerable fraction of the pollution load to surface water. A CSA is where pesticide sources intersect with areas of high mobilisation potential which have the highest propensity for surface runoff generation, pollutant transport and delivery via hydrologically connected pathways (Doppler et al., 2014, 2012; Frey et al., 2009). The spatial variability of pesticide losses to waterbodies can be significant (Freitas et al., 2008; Leu et al., 2004a, 2004b) therefore the identification of CSA will help target mitigation measures efficiently to locations where they can strongly reduce pesticide loads into river courses.

The spatial variability within a catchment, e.g. different soils and land uses, different travel times and flow lengths from each parcel to the catchment outlet and different application dates of pesticides increase the complexity and the variables that must be included to determine where mitigation measures should be proposed. Topography (which governs the flow paths of surface water) and the position of landscape elements such as riparian buffer strips, grassed waterways, hedges, ditches, decisively influence if and what fraction of applied pesticide ultimately reaches a watercourse (Reichenberger et al., 2007). The amounts of pesticides reaching water resources vary considerably in time and space and are highly dependent upon application rates and the chemical characteristics of the pesticides, as well as soil and climate conditions (Doppler et al., 2014, 2012; Freitas et al., 2008; Leu et al., 2004a).

Assessment and identification of areas contributing to non-point source (NPS) pollution by pesticides has been performed in other approaches using hydrological models to approximate contaminant transport (Bach et al., 2002; Lescot et al., 2013; Wohlfahrt et al., 2010), a combination of indicators and multi-criteria analysis (Macary et al., 2014), GIS modelling to prioritize catchments or streams within a watershed (Zhang et al., 2008), or the use of long-term pesticide monitoring data (Di Guardo and Finizio, 2018). These approaches were applied mainly to larger scales (watershed) to identify risk zones. Also, these studies do not consider the microscale required for recommendations within a small catchment. Although the watershed scale is proper to achieve environmental goals for water quality, changes in agricultural practices and the implementation of mitigation measures like field buffer strips take place at field level (McGonigle et al., 2012). Therefore, risk assessment at field scale is useful when the implementation of actions by farmers is needed (Bereswill et al., 2014).

A range of management techniques is available to control agricultural pollutants such as the reduction of pesticide use and the installation of landscape features like buffer zones, hedgerows, retention ponds and wetlands that can capture and degrade pollutants before they reach watercourses (Bereswill et al., 2014; Reichenberger et al., 2007). The use of mitigation measures that minimise the risk of off-site pesticide pollution caused by spray drift, drain flow and runoff could contribute to achieve the good status of water bodies (Aguiar et al., 2015; Maillard et al., 2012).

We propose a robust and spatially explicit model-based (Mb) risk approach to identify priority areas to target landscape mitigation measures in order to reduce pesticide pollution and erosion in surface water. The Mb risk relies on geospatial emission modelling and connectivity of parcel sites towards waterbodies. The impact of crop rotation during five-years is analysed for this catchment. The Mb risk method is applied in a case study in the southeast of Flanders, Belgium. The Mb risk areas are then compared with an observation-based (Ob) approach that includes field observations for relevant processes identified for this catchment by local experts.

2. MATERIALS AND METHODS

2.1. Site description

The catchment for this study is located in Sint-Truiden, SE Flanders, Belgium (Figure 1). The site has an area of 10,7 km^2 with altitudes ranging from 51 to 107 m above sea level. The Cicindria river flows from South to North with a length of 6.5 km within the catchment limits. The dominant land use is agriculture covering 72% of the area; 32% with fruit trees (apple, pears and cherries) and 68% arable crops (cereals, beets, maize mainly).

The area is characterised by a hilly topography with slopes of up to 32 degrees and loamy soils, resulting in a high vulnerability to erosion and muddy floods (Evrard et al., 2007). Soils are well-drained and usually have no artificial drainage systems. The mean annual precipitation varies between 700 and 900 mm with a calculated erosion potential of 6,7 ton ha^{-1} year^{-1} (Bureau for Environment and Spatial Development – Flanders). High erosion and muddy floods, an extensive area under agricultural use and relatively high measured concentrations of pesticides in surface water were reasons behind the site selection.

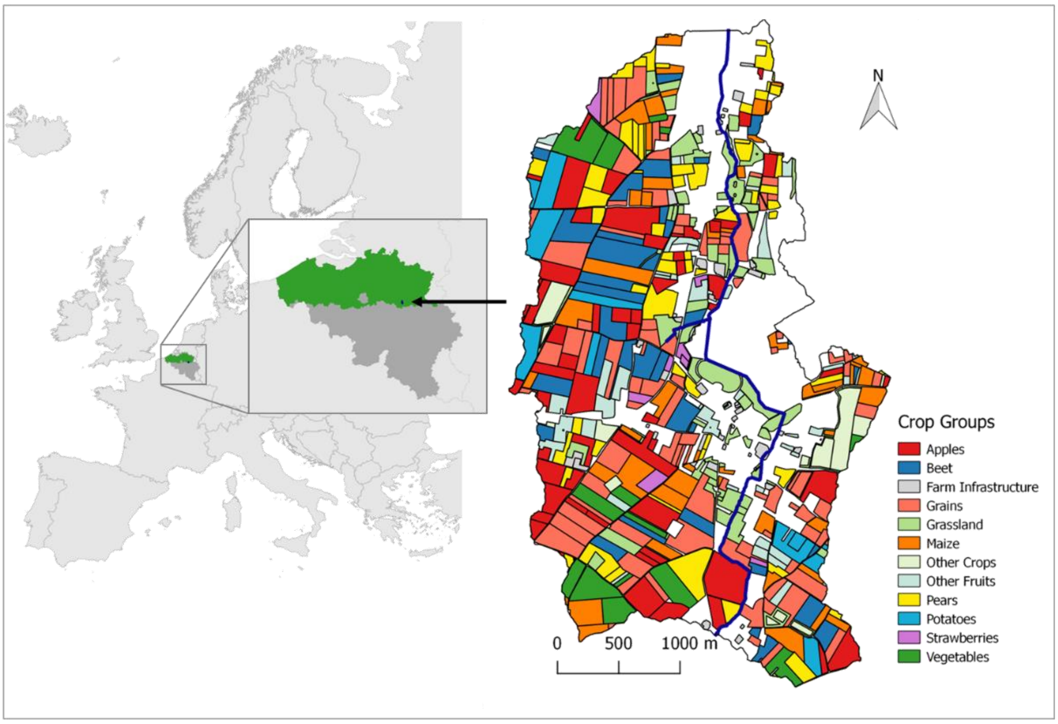


Figure 1. Cicindria Catchment: Location and land use (2012)

2.2. Active Ingredient

Although the general approach for delineating the critical zones is generic to any active ingredient, we illustrate the approach here for glyphosate. This herbicide is extensively used worldwide and intensively applied to agricultural fields (Benbrook, 2016). It is frequently detected in agricultural basins where the herbicide is applied (Battaglin et al., 2014; Chang et al., 2011; Coupe et al., 2012; VMM, 2015).

Glyphosate is a broad-spectrum, non-selective herbicide used in a wide range of crops, either before planting, pre- or post-harvest. It is applied in both conventional and reduced/no-till farming to control the growth of annual and perennial weeds.

2.3. Model-based (Mb) risk: Potential NPS glyphosate risk map from geospatial modelling

The Model-based (Mb) risk map for pesticides exports to surface water bodies is obtained considering the different pathways that pesticides can follow after application towards the surface water. Based on geospatial emissions and calculated connectivity, the Mb risk map is a theoretical approach, computed at field scale for a catchment area. Figure 2 is a stepwise description of the methodology including the datasets required.

2.3.1 Step 1: Geospatial emission modelling

Emission modelling is based on the WEISS approach (http://weiss.vmm.be/) using emission factors (EF). EFs represent the fraction of the total applied product that goes through the different pathways allowing the distinction between the transport routes. The amount of the substance emitted per year can be calculated using EFs. The contribution of the different transport pahtways, in this case drift, volatilisation, erosion and drainage, were calculated using EFs. These factors can be estimated based on methods from literature (De Schampheleire et al., 2007; Linders et al., 2000; Pussemier, 1999; Webb et al., 2016). Each emission pathway was estimated separately using available datasets (Table 1). The gross emissions to surface water are the sum of drift, erosion and drainage considering, only diffuse pollution. Point losses are not included in this study.

The dose of pesticide is spatially assigned to a specific field using a detailed land cover (LC) map. The LC map was reclassified initially in 11 crop groups (Table 2) defined by the Department of Agriculture and Fisheries (DLV: Departement Landbouw en Visserij) and Department of Monitoring and Study (AMS: Afdeling Monitoring en Studie) in Belgium (Van Esch et al., 2012). As shown in Table 2, each LC has a pesticide dose that corresponds to the average dose per cultivation reported for Flanders obtained from different departments of the Belgian agricultural administration, including the Agricultural Monitoring Network (LMN: Landbouwmonitoringnetwerk), DLV and AMS (Van Esch et al., 2012).

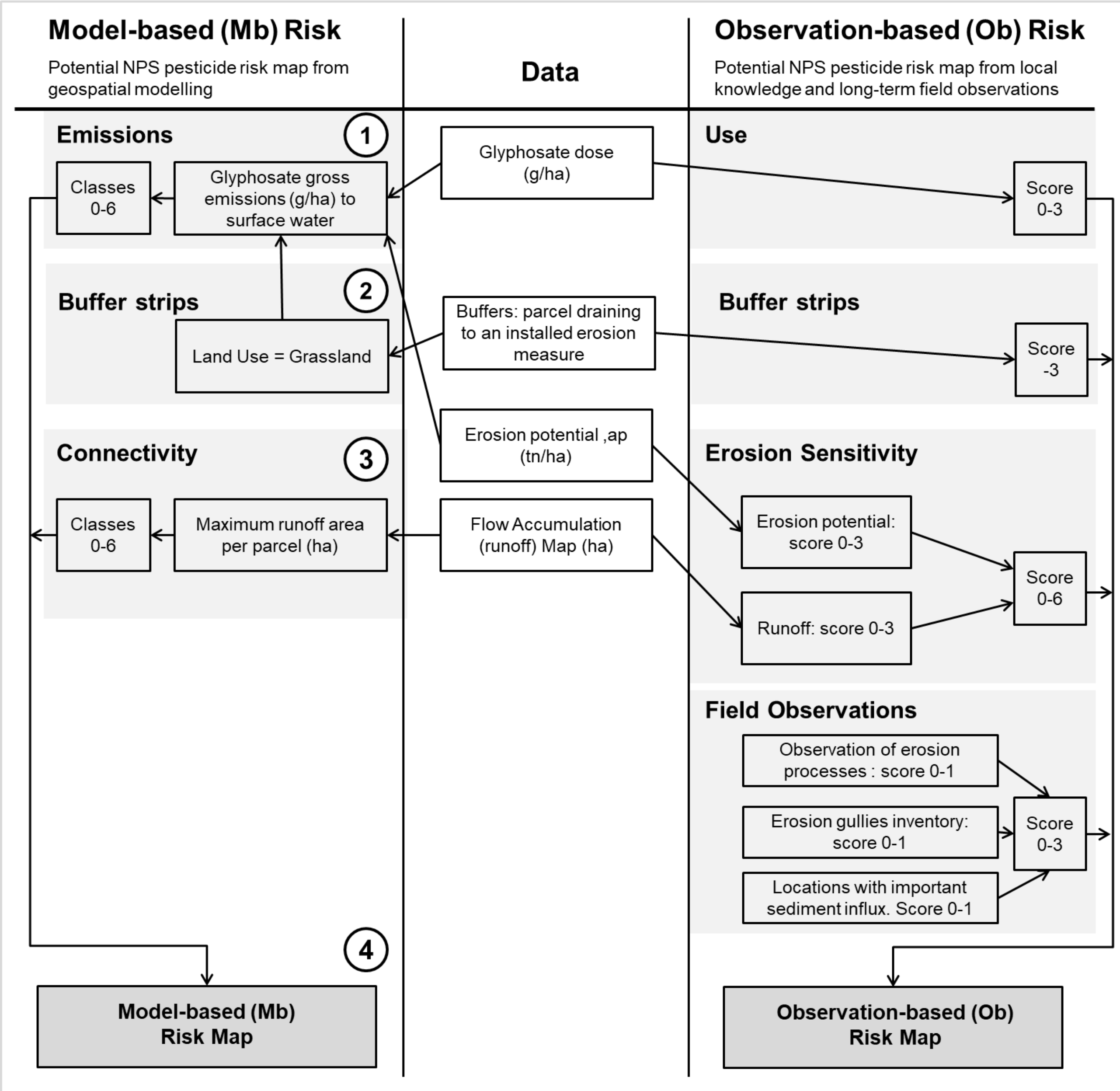


Figure 2 Flowchart with the methodology developed for this study. Model-based (Mb) risk map and Observation-based (Ob) risk map are two approaches by using similar datasets can evaluate potential risk for pesticide surface water pollution to target priority areas. The figure shows the four steps for the Mb risk map method: 1) emissions 2) buffer strips 3) Connectivity 4) Mb risk map. Ob risk method is obtained from the compilation of scores that includes field observations and expert local knowledge.

Table 1 GIS databases: Input data and sources

Data	*Type of data*	*Resolution*	*Source*
Land cover data 2008-2012	Shapefile	1:2000	Department of Agriculture and Fisheries www.geopunt.be
Flow accumulation from multiple flow direction (MDF)	Raster	5m	www.geopunt.be
Erosion Potential	Shapefile		Bureau for Environment and Spatial Development - Flanders
Watercourses	Shapefile		www.geopunt.be
Mitigation Measures	Shapefile		VLM – Flemish Land Agency
Soil map (Bodemkaart van Vlaanderen 2001)	Shapefile		Flemish Land Agency

Table 2. Main crops and average glyphosate dose reported per cultivation (source: LMN, DLV, AMS) in each crop group

Crop Group	*Crops*	*Annual glyphosate dose (g ha^{-1})*
Potatoes	Potatoes	380
Strawberry	Strawberries	1390
Vegetables	Peas, beans, brussels sprouts, spinach, leek, chicory	1240
Apples	Apples	1080
Pear	Pears	1080
Other Fruits	Raspberries, blackberries, blueberries, cherry	800
Grains	Oats, spelt, winter wheat, summer wheat, winter barley, winter rye, winter oilseed rape	380
Maize	Maize grain, maize fodder	250
Beet	Fodder beet, sugar beet	330
Grassland	Permanent grassland, temporary grassland, grass mixture, pasture with trees, natural grassland with minimum activity	250
Other Crops	Annual grass-clover, perennial grass-clover, other fodder crops, other afforestation, flower mixture	440

The EFs for the different transport pathways were calculated for the different crop groups following the procedure explained below.

Drift is the amount of pesticide that is deposited beyond the boundaries of the treated area by air currents and depends upon the spraying equipment, meteorological conditions and growth stage of the crop. The German Ganzelmeier curves were used to predict drift at certain distances downwind of the field with a distinction between the categories orchard and field crops (De Schampheleire et al., 2007). An average EF drift is allocated to each field that has a border with a watercourse. The EF drift applies only to the 10 m area of each field closest to the river. The average drift is calculated from the total drift in a 10 m zone around the watercourse distributed evenly across the zone. This results in an EF drift of 0.013 and 0.324 for field crops and fruit applications, respectively.

Volatilisation is the fraction of the applied dose entering the atmosphere through volatilisation after application. EF Volatilisation is determined by the vapour pressure of the pesticide and the classification proposed by the European Environment Agency (EEA) in the air pollutant emission inventory guidebook (Webb et al., 2016; Woodrow et al., 2001). Glyphosate is a low-class volatile substance (Table 3) according to the classification shown in Table 4.

Table 3 Glyphosate properties required for the emission factors

Properties	*Glyphosate*
Formula	$C_3H_8NO_5P$
Vapour pressure at 25°C (mPa)	0.0131[a]
Half-degradation time, DT50 field (day)	3-174. Typical 15[a]
Soil organic partition coefficient Koc (L kg^{-1})	884-50660[a] Typical 16331

[a] Source: Lewis et al., 2016 - PPDB: Pesticide Properties DataBase

Table 4 Volatilisation emission factor for pesticides from their vapour pressures (from Webb et al., 2016)

Vapour pressure class	*Vapour pressure, mPa*	*Emission factor*
Very high	P > 10	0.95
High	1 < p < 10	0.50
Average	0.1 < p < 1	0.15
Low	0.01 < p < 0.1	0.05
Very low	p < 0.01	0.01

Foliar Interception represents the fraction of the applied product that has contact with the foliage. The interception emission factors for the different crops were obtained from Linders et al. (2000) using the information on the time of application and the development stage of the crop. Data from product registration Fytoweb (www.fytoweb.be) was used to determine the development stage of the cultivation in which the substance is typically applied.

The losses of glyphosate due to **erosion** were calculated indirectly from deposition, which is the fraction of the applied pesticide that reaches the soil. It was assumed that a pesticide is homogeneously mixed in the top layer considered to be 3 cm of soil. The amount of pesticide present in the top layer of the soil is then multiplied by the potential water erosion estimated for each field. Applied pesticides are initially concentrated in topsoil layers (Okada et al., 2016; Rampazzo-Todorovic et al., 2010; Yang et al., 2015). The content of the pesticide in the upper layer is calculated through deposition (Eq. 1):

$$Pesticide\ soil\ deposition\left(\frac{kg}{ha}\right) = applied\ dose\left(\frac{kg}{ha}\right) - drift\left(\frac{kg}{ha}\right) - volatility\left(\frac{kg}{ha}\right) - interception\left(\frac{kg}{ha}\right) \quad (1)$$

The weight per hectare of the upper 3 cm of soil top layer was calculated considering the soil density of the soil according to the soil texture classes for Flanders (Eq. 2). The study area has mainly loam soils with an average soil density of 1400 kg m^{-3}:

$$G\left(\frac{ton}{ha}\right) = soil\ density\ \left(\frac{kg}{m^3}\right) \times 10000\ \left(\frac{m^2}{ha}\right) \times\ 0.03\ m \quad (2)$$

The deposition (g ha^{-1}) is divided then by the weight of the upper layer to obtain the grams of glyphosate per ton of soil (Eq. 3):

$$B(\frac{kg}{tn}) = \frac{Soil\ deposition\ (\frac{kg}{ha})}{G\ (\frac{tn}{ha})} \quad (3)$$

Finally, the emissions from soil erosion are obtained by multiplying the deposition (g ton^{-1}) by the potential water erosion occurring on each specific parcel (Eq. 4). The potential water erosion map was modelled each year by the Bureau for Environment and Spatial Development (Flanders) applying the RUSLE equation (Kinnell, 2010; Renard et al., 1997). The C factor (crop factor in RUSLE) was adjusted according to land cover (Table 5). The Erosion calculation is based on estimates of potential erosion and therefore does not consider possible implemented mitigation measures limiting the amount of sediment that effectively reaches the watercourse.

$$\text{Pesticide from Erosion}\left(\frac{kg}{ha}\right) = B\left(\frac{kg}{tn}\right) \times potential\ erosion\left(\frac{tn}{ha}\right) \tag{4}$$

Table 5 C factor for different crop groups

Crop Group	*C-factor*	*Source*
Potatoes	0.49	Average of potato for storage and early potato (Ruysschaert, 2005)
Strawberry	0.28	Similar to maize and beet
Vegetables	0.50	Average chicory, early leek, late leek, carrot, peas/beans, onion, shallot, early cauliflower and late cauliflower (Ruysschaert, 2005)
Apples	0.05	Fruit trees (Verbist et al., 2004)
Pear	0.05	Fruit trees (Verbist et al., 2004)
Other Fruits	0.28	Similar to maize and beet
Grains	0.15	Average of winter wheat, winter barley, summer wheat, summer barley and oats (Ruysschaert, 2005)
Maize	0.28	Average of grain and fodder maize (Ruysschaert, 2005)
Beet	0.28	Average of sugar beet and fodder beet (Ruysschaert, 2005)
Grassland	0.08	Average of permanent grass, temporary grass (Verbist et al., 2004)
Other Crops	0.29	Average of chicory and flax (Ruysschaert, 2005)

Pesticides with a high affinity to soil sorbents are likely to migrate in the particle-associated form (Gevao et al., 2000). Erosion is considered as the primary loss pathways for strongly sorbing substances with high K_{OC} (soil organic carbon partition coefficient) with values greater than ca. 1000 L kg^{-1} (Reichenberger et al., 2007; Tang et al., 2012; Wu et al., 2004).

Drainage was determined by the GUS number (Groundwater Ubiquity Score) (Gustafson, 1989; Pfeiffer, 2010). GUS is a function of K_{OC} and DT50 (half-life) in the soil for the applied substance. Both properties were obtained from Pesticides Properties DataBase (PPDB) indicated in Table 3 (Lewis et al., 2016).

$$GUS = \log_{10} DT_{50} \times (4 - \log_{10} K_{oc}) \quad (5)$$

EF Drainage for a given pesticide corresponds to a particular class from GUS following the classification proposed by Pussemier L. (1999) shown in Table 6.

Table 6 Drainage emission factor from Pussemier(1999)

GUS	*EF Drainage*
GUS < 3	0.0001
3 < GUS < 4	0.001
4 < GUS < 4.5	0.01
GUS > 4.5	0.1

The sum of drift, erosion and drainage losses represents the gross emissions towards the surface water.

2.3.2 Step 2: Buffer strips

Vegetated buffer strips reduce the speed of the runoff water and thus also the transport capacity and leading to sedimentation of the dissolved soil particles (Muscutt et al., 1993). They can be used to reduce pesticide input to surface waters (Syversen and Bechmann, 2004). The efficiency of buffers for pesticides depends on the width, cover, location, pathway and the properties of the substances. Therefore, efficiencies ranges vary widely (Lerch et al., 2017; Pätzold et al., 2007; Reichenberger et al., 2007).

The study site had buffer strips established on several fields at the time of the evaluation (2012). The information about the location of these elements was possible to include in our analysis and, this allowed for the comparison with the results from the observation-based method. The parcels which drain to a buffer were identified. In order to keep our calculations simple, we reclassified these fields as grasslands, and then, the emissions from these fields were recalculated.

2.3.3 Step 3: Connectivity and runoff area

The hydrological connectivity of each parcel is evaluated using the runoff upslope contributing area map. The runoff map shows the zones that can potentially produce runoff to the location of interest (i.e., a grid cell) and is a static representation of runoff generation (Bracken and Croke, 2007). The lines in Figure 3 (left) represent the hectares of land from which the runoff flows over a cell. The values are cumulative, and increasing cell values indicate the main runoff route of the water.

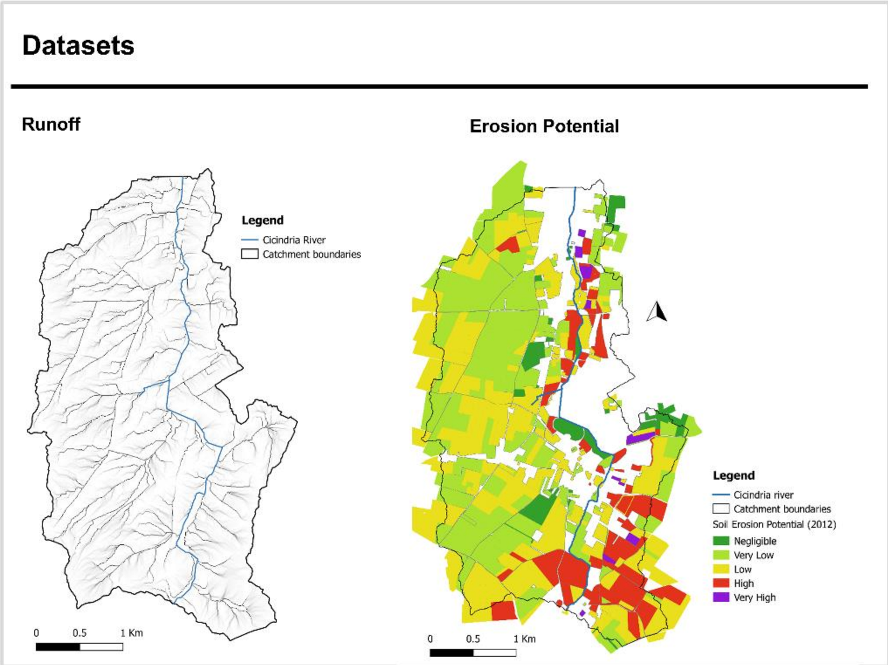


Figure 3 Datasets. Left: Runoff or flow accumulation map. Right: soil erosion potential calculated for 2012

We used a flow accumulation or runoff map provided by the Department of Land and Soil Protection, Subsoil, Natural Resources (ALBON: afdeling Land en Bodembescherming, Ondergrond, Natuurlijke Rijkdommen). A multiple flow direction (MDF) raster file was the initial input for the runoff map based on a Digital Elevation Model (DEM) with 5 m resolution and the Flemish Hydrographic Atlas. It considers the topography and the existing watercourses. The multiple streamlines indicate where the water flow potentially concentrates after rain. The runoff raster file is overlaid with the agricultural parcels or land cover maps (polygon shapefile). The highest raster cell value within a parcel polygon represents the runoff area connected to this parcel, and this maximum value is assigned to each parcel to generate the connectivity map. The more extensive the runoff over a cell, the higher the importance of the cell.

2.3.4 Step 4: Model-based (Mb) risk map

The gross emissions (g ha^{-1}) and the connectivity (max ha runoff per parcel) were calculated for each agricultural parcel and then classified using Table 7. The class numbers increase with increasing emissions and with increasing connectivity, i.e. increasing of connected runoff area. These two maps, emissions and connectivity, were added to obtain the Mb risk map (max 12 classes). In summary, to each agricultural field we assigned a class number for the gross emission of glyphosate; corrected in case buffer strips were installed, and another one for connectivity to the river.

Table 7 Glyphosate gross emission and connectivity classes

Classes	*Glyphosate gross emissions (g ha^{-1})*	*Connectivity (Runoff Ha)*
1	0-0.6	<1
2	0.6-1	1-2
3	1-1.5	2-5
4	1.5-2	5-20
5	2-4	20-50
6	>4	>50

2.4. Temporal evaluation

The impact of crop rotation on risk areas was investigated. The Mb risk methodology was applied over five years (2008-2012). Each resulting Mb risk map was further classified to consider only high-risk areas. Fields with a total risk equal to or higher than 9 (up to 12) were considered areas to prioritise with a higher risk for pesticide pollution. The maps were added to a cumulative map for the five years.

2.5. Observation-based (Ob) risk map: Potential NPS glyphosate risk map from local knowledge and field observations

For this study site, an alternative approach was performed using similar datasets. The Observation-based (Ob) risk map incorporates long-term field observations and local expert knowledge. Four factors were combined into one aggregated risk score per parcel: glyphosate use, erosion sensitivity with scores for erosion potential and runoff, the presence of mitigation measures and field observations as shown in Figure 2. A score for each of the four factors was assigned to each agricultural field. Average glyphosate dose for arable crops (550 g/ha) and fruit trees (1080 g/ha) was considered. The score for glyphosate use is shown in

Table 8.

Table 8 Scores for glyphosate use

Average dose (g/ha)	*Score*
0	0
0-500	1
500-1000	2
>1000	3

Potential water erosion scores were assigned using the potential water erosion map (Table 9). The flow accumulation map (Figure 3, left), previously used in the Mb method, was used to find the maximum runoff area per parcel to assign a score for the subfactor runoff (Table 10).

Table 9 Scores for subfactor potential water erosion

Potential Water Erosion	*Score*
Negligible (dark green) and very low (light green)	1
Low (yellow) and medium (orange)	2
High (red) and very high (purple)	3

Table 10 Scores for subfactor runoff

Runoff Area (ha)	*Score*
0	0
0-10	1
10-50	2
50	3

Parcels that drain to an implemented buffer strip were identified. As shown in Figure 2, these plots received a score of -3, thus reducing the eventual glyphosate exports towards waterbodies. Three types of field observations were considered: observation of erosion processes, erosion gullies and sediment influx into the river (Figure 4). The parcels received a score of 1 for each of the observations.

Figure 4 Field observations from the study site A) Erosion observations B) Erosion gullies C) locations with significant sediment influx towards the river.

3. RESULTS AND DISCUSSION

3.1. Model-based risk areas

Our purpose was to develop a desktop analysis to identify critical areas that could be included in a mitigation action plan to reduce pesticide loads into surface water. The approach considers the potential pesticide emissions and the hydrological connectivity of each parcel. Figure 5 shows the resulting Mb risk map for 2012.

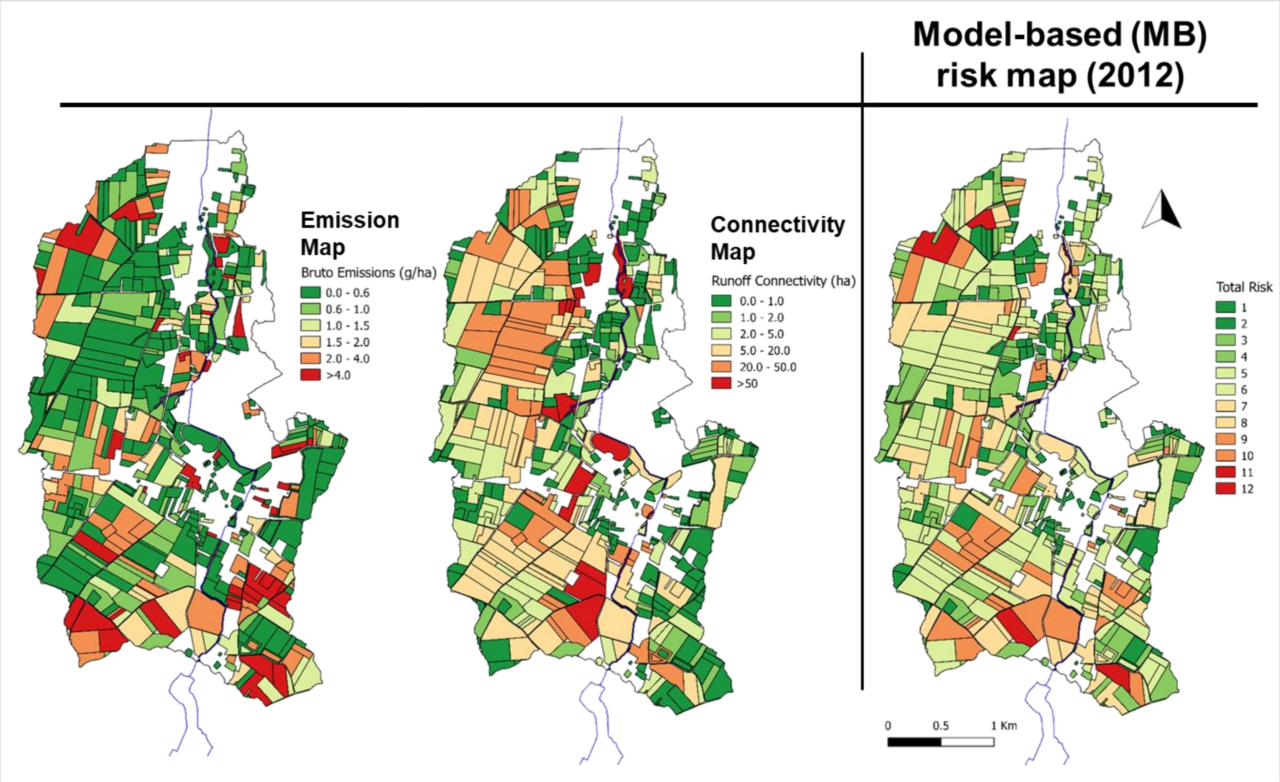


Figure 5 Right: Mb risk map for 2012 with 12 risk classes obtained by the sum of the two intermediate maps, emission (left) and connectivity (centre).

The Mb risk map allows the identification and prioritisation of specific fields within the catchment. Once the critical areas are identified, the within-field hotspots need to be determined to set appropriate locations for the implementation of additional landscape mitigation measures. The hydrologic pathways from the runoff map used for connectivity may help to identify which channels are good candidates for landscape measures such as grass buffer strips, dams or detention basins.

The Mb risk methodology can be easily implemented in other catchments as long as datasets are available for the evaluation. For instance, this approach could be applied in other catchments in Flanders (Belgium), since datasets are readily available. Moreover, the methodology is generic enough to be used for other pesticides. Based on the properties of the substance, other emission pathways could be more prominent.

Runoff could contribute to a variable amount of pesticide depending on the strength and timing of rainfall events (Doppler et al., 2014). However, the calculation of the emissions did not consider the fraction of dissolved pesticide that reaches the river through runoff events. Nevertheless, runoff was not completely neglected as it was included in the connectivity score.

The results cannot be used to predict pesticide concentrations or loads of pesticides. The approach provides a static snapshot of the potential annual risk according to specific land use of the area. Some dynamic parameters such as soil moisture, rainfall intensity, and the application time of the pesticide were not considered in the evaluation.

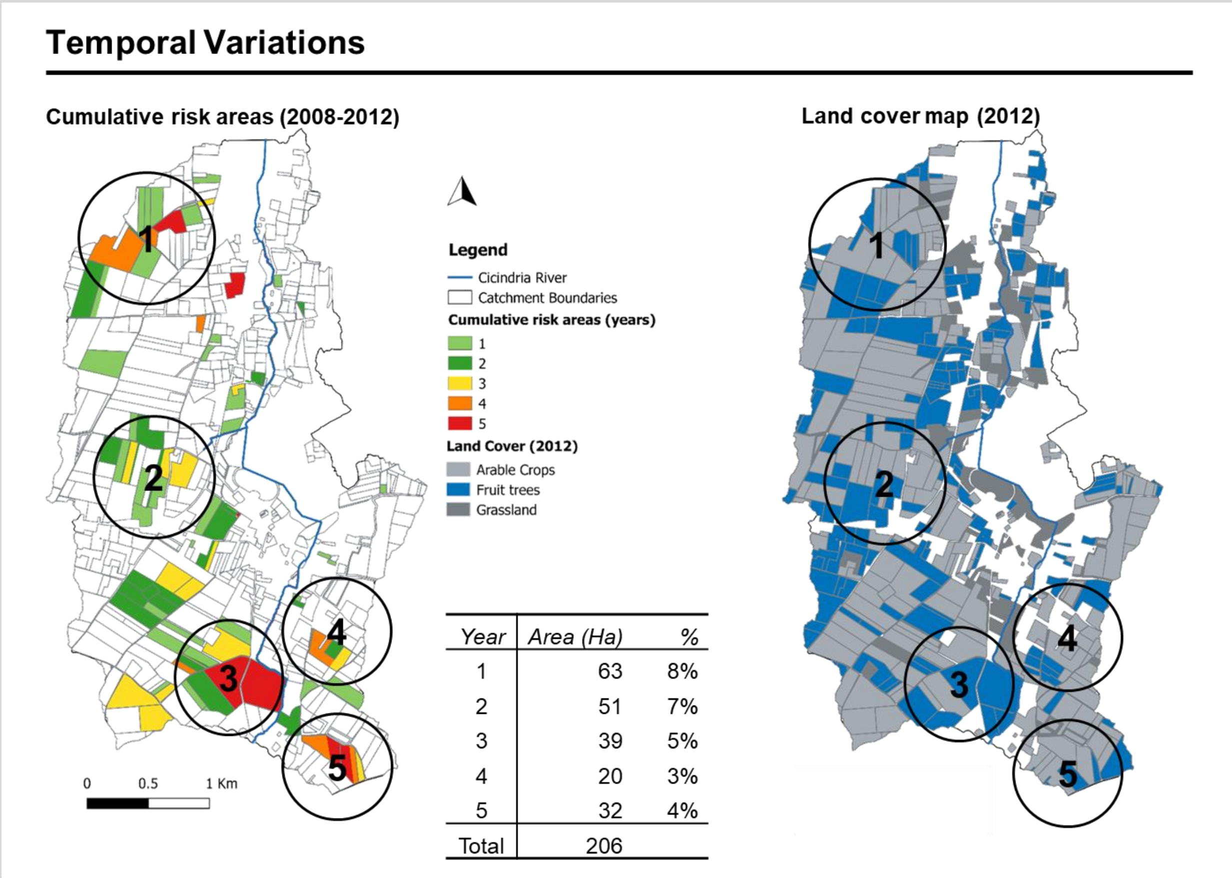
We opted for a practical approach treating the whole parcel as grassland when the parcel had a buffer strip implemented at the time of the evaluation. By doing this, emissions from these parcels were reduced on average by 95%. In comparison with the literature, the simplification may show an overestimation of the efficiency of the buffers (Lerch et al., 2017; Reichenberger et al., 2007; Syversen and Bechmann, 2004; Zhang et al., 2010) leading to an underestimation of risk when parcels with buffer strips are evaluated. A more detailed modelling process considering the characteristic of the buffers strips is required in order to reduce the discrepancy between these results and the ones obtained from empirical data.

3.2. Temporal evaluation

Grassed buffer strips in the study area are implemented under voluntary five-year contracts between the farmers and the government. Crop rotation over the years could induce changes in the potential risk of pesticide exports. The Mb risk approach was applied for several consecutive years (2008-2012) to explore temporal variations. A cumulative map (Figure 6) compiles the areas with high risk during those five-years.

Figure 6 indicates the years that a particular area had high-risk score (over or equal 8). Out of the total agricultural area of the catchment (773 ha), 26% (206 ha) show a high potential risk for glyphosate for at least one year of the studied period. Only 4% (32 ha) exhibit high risk on each of the five years, disregarding the crop planted.

The cumulative map is compared with a land cover map (2012) to further analyse the type of crop in the areas with permanent risk (Figure 6). Interestingly, this study site has a considerable proportion covered by fruit trees which is a permanent crop type. In zone 3 mainly orchards are present and grass is permanently maintained between the tree rows to avoid soil erosion. Therefore, additional buffer strips are not necessary for zone 3. In the other zones circled in Figure 6 where arable crops are part of the rotation, grass buffer strips could be suggested to mitigate pesticide pollution. This temporal evaluation could help risk managers to prioritise fields for the implementation of mitigation measures in consideration of crop rotation and long-term analysis.



Year	Area (Ha)	%
1	63	8%
2	51	7%
3	39	5%
4	20	3%
5	32	4%
Total	206	

Figure 6 Left: Cumulative higher risk areas 2008-2012 (5 years). Right Land cover map (2012) to seek for permanent crops (fruit trees) and arable rotated crops. A table (middle) shows the cumulative surface (ha) identified as a high-risk area during the five years.

3.3. Ob risk map: field observations

The distinctive aspect of the Ob map is the field observations that include aspects not easily evaluated with a model. Figure 7 shows the maps for the three types of observations recorded from the study site. The sediment influx map covers human-made networks like roads that influence the hydrology of the catchment. The parcels that contribute to a higher amount of sediments that could enter directly into the river from a road were identified. Previous studies demonstrated the relevance of these elements for water quality (Carluer and Marsily, 2004; Hösl et al., 2012). The long-term field observations included in these maps can capture processes such as fast run-off over roads and observed sediment influx to the river, as well as erosion gullies, that are difficult to obtain as a result of modelling approaches.

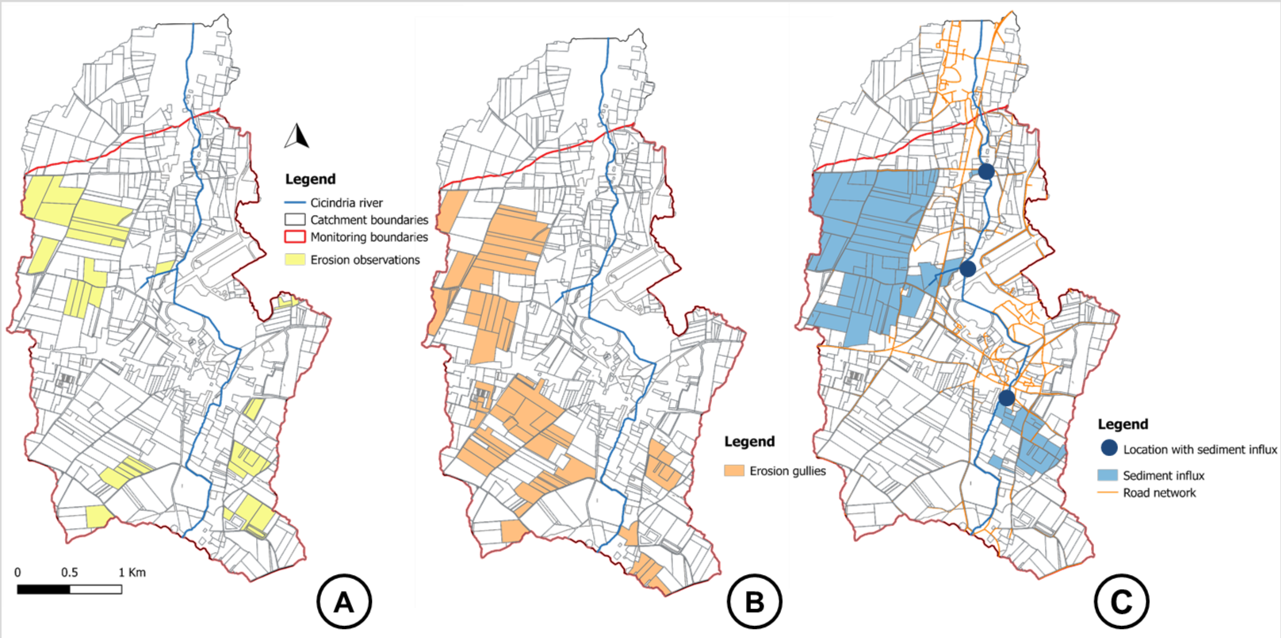


Figure 7 Field observations Left: A) Erosion observations. Center: B) Erosion gullies inventory. Right: C) locations with significant sediment influx towards the river. Roads in some parts of the catchment behave as pollutant pathways when high-rainfall events occur.

3.4. Model-based risk vs Observation-based risk map

We compared the model-based risk map for one year (2012) with the observation-based risk map (Figure 8). The field observations were performed in a smaller area within the catchment. For that reason, some fields in the northern part of the catchment downstream, ahead of the monitoring station Sint Truiden, could not be compared. However, we can observe regions where both methodologies identify critical zones (Figure 8). Though, not all the parcels within these regions matched with the same risk. Field observations considered in the Ob risk method capture processes that are not included in the Mb risk map, such as the sediments that flow in some roads during intense rainfall or parcels with evidence of erosion processes that are not detected in the simulated erosion potential map. Particularly for this catchment, the road network plays an essential role in connectivity, and roads facilitate the water flow with sediments during certain rainfall events.

Both approaches were congruent in identifying broader areas of risk as to the circled areas in Figure 8 show. These areas could help with the delineation of critical zones. Nevertheless, they differ in a field by field comparison.

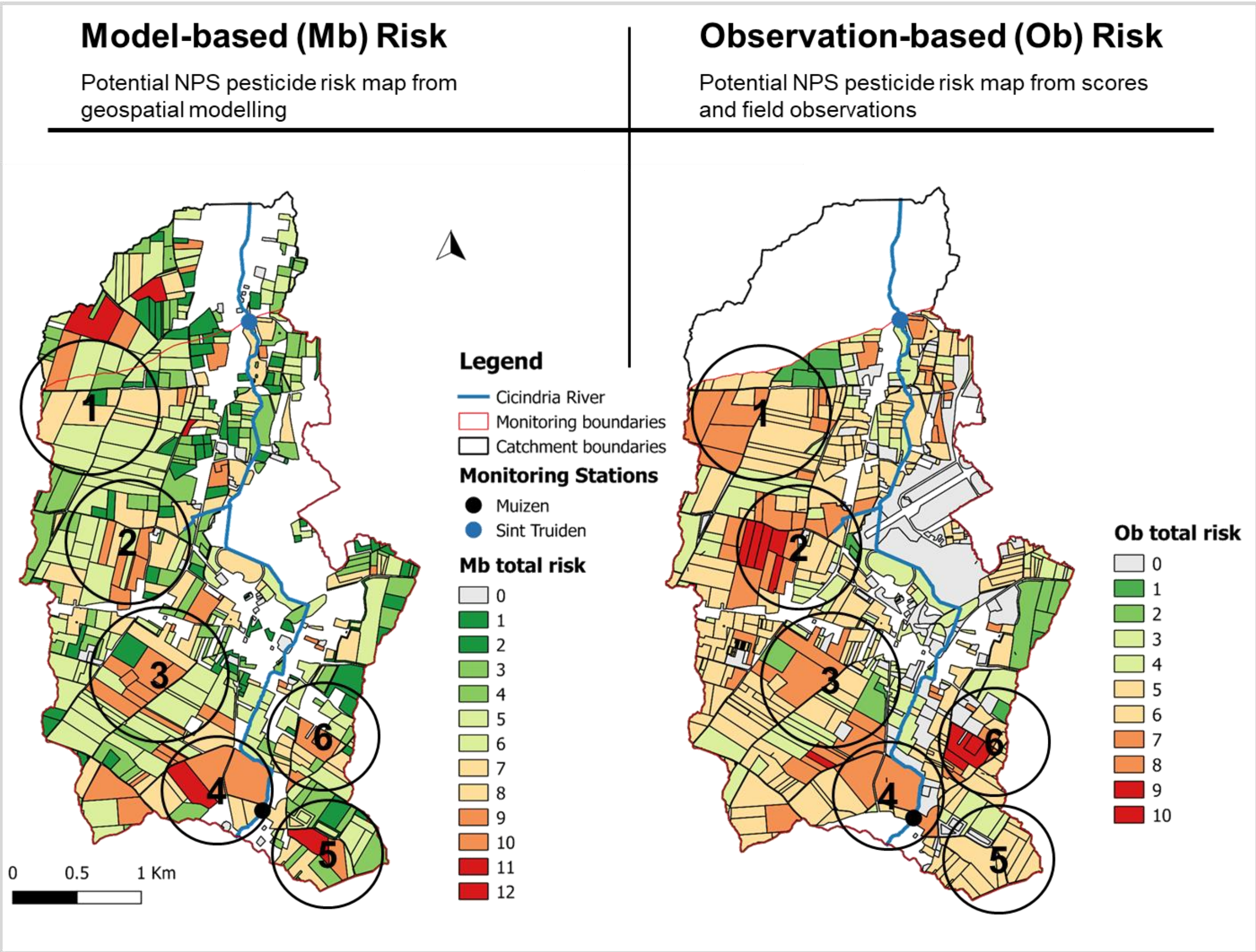


Figure 8 Left: Model-based risk map (2012). Right: Observation-based risk map (2012)

Six critical areas could be easily identified to compare the two approaches (Figure 8). The risk in zone 2 is much higher in the Ob risk map than in Mb risk map because the parcels in this zone showed evidence of erosion processes, erosion gullies and sediment influx. Buffer strips were not implemented in zone 2. Erosion gullies were observed in zone 3 and 4 which provides evidence of long-term soil erosion. Zone 5 is an area with high erosion sensitivity partly because of the presence of crops with a low-c factor and high connectivity to the river. Therefore, the total risk evaluated with the Mb risk method is higher than from the Ob risk approach. The risk in zone 6 is much higher in the Ob risk method which includes field observations. Erosion gullies, as well as fast runoff and sediment influx over the adjacent roads in the parcels, can be observed in zone 6.

Table 11 Risk classes

	Mb Risk	*Ob Risk*
Low	1-4	1-3
Moderate	5-8	4-6
High	9-12	7-10

In order to compare spatial patterns and the level of agreement, each map was aggregated in three classes (low, moderate, high) following Table 11. After the classification, the coincident areas per class from both maps were examined. Figure 9 shows the concurrent areas where both methods classified these parcels as high, as well as the non-coincident fields. The level of agreement between both approaches is shown in Table 12. In green is the coincident area classified by both methods for the three risk classes. 50% of the agricultural land was classified in the same class, and only 4 ha has a distance of two classes (1 ha low Ob – high Mb, 3 ha high Ob-low Mb). The agreement was considered satisfactory for this study considering that the two approaches cannot be readily compared. The Mb risk method requires less time for the evaluation, and it is based on available datasets. Therefore, it can be recommended for initial screening of risk zones. Nevertheless, site inspection of the priority areas is essential afterwards when individual farms need to be targeted.

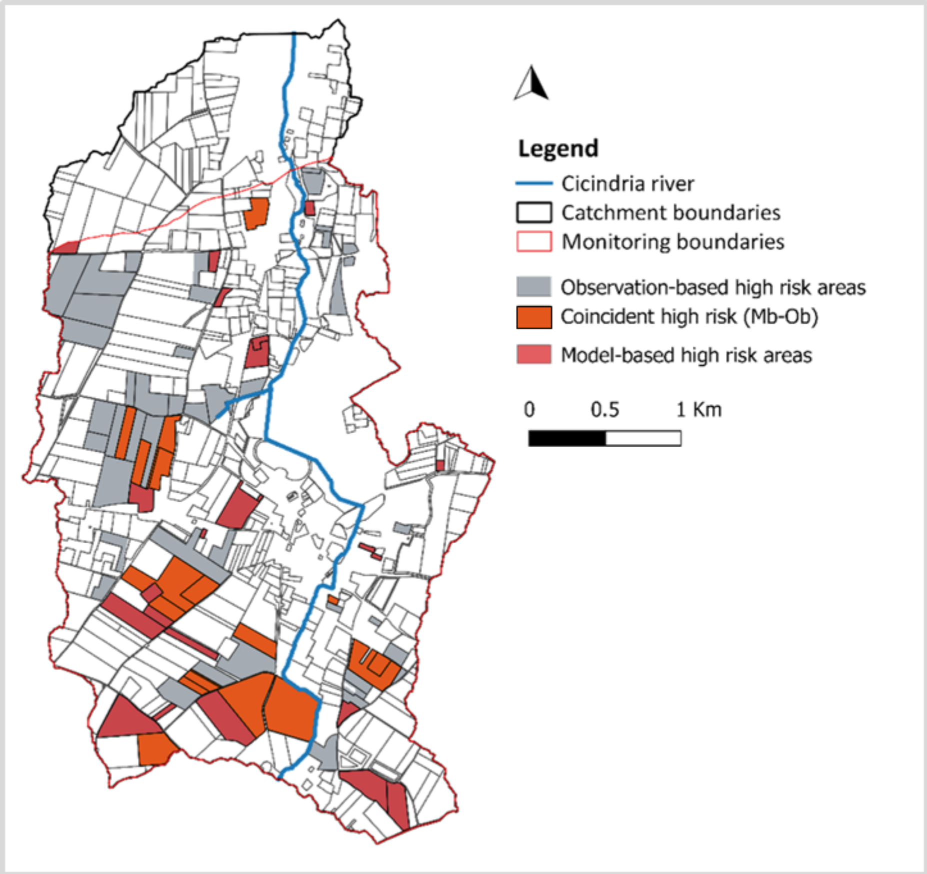


Figure 9 Mb risk map vs Ob risk map for 2012: comparison of the resulting maps obtained from both methods.

Table 12 Comparison between Model-based (Mb) and Observation-based (Ob) approach

				Model-based risk areas				
				180	382	111	673	Total Mb Area
				27%	57%	16%		
			(values in ha)	Low	Moderate	High		
Observation-based risk areas	108	16%	Low	**56**	**49**	**1**	106	52%
	382	59%	Moderate	**114**	**218**	**46**	378	58%
	163	25%	High	**3**	**100**	**60**	163	37%
	653 Total Ob Area			172	367	108	**334**	50%
				32%	59%	56%		

4. CONCLUSIONS

We developed a GIS-based tool for water resource managers to help in the identification and prioritisation of critical source areas. The tool is relatively simple to apply and uses geospatial data that it is often relatively accessible. We propose the model-based risk method as a valuable approach to detect priority areas for actions against diffuse pesticide pollution. It identifies areas in which mitigation measures seem necessary and could, therefore, contribute to improving water quality.

The method was not envisioned to predict pesticide concentrations. Moreover, emissions were not empirically validated. It was developed for a relative comparison of the parcels to identify priority zones.

We tested the Mb risk method in the Cicindria catchment. Results from the case study showed a good performance in comparison with an observation-based method which is a more time-consuming methodology and requires long-term field observations and expert knowledge of the site. The use of detailed crop classes allows a crop rotation analysis and the evaluation of temporal changes.

The results obtained in this study were used to identify, raise awareness about and motivate farmers to voluntary implement mitigation measures such as grassed buffer strips. It is an advantageous tool to explain to farmers the source and pathways of pesticides and to discuss the proper location of risk reduction measures. To continue these investigation efforts, the effect of these measures on the glyphosate loads in the river is being assessed by a long-term monitoring campaign of five years (2014-2018).

The management of CSAs may be an effective measure to reduce agrochemical exposure of streams in agriculture catchments. The methodology proposed in this paper allows a first picture of the potential impact of pesticides on surface water bodies; it detects risk areas and assesses the priority of the actions that should be undertaken as part of the catchment management strategy.

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